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River dikes in agricultural landscapes: the importance of secondary habitats in maintaining landscape-scale diversity

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Abstract

Lowland rivers and their floodplains have changed markedly over the last centuries. River dikes have become among the most extensive secondary habitats of former floodplains. Our main question was, what role do secondary habitats on river dikes play in harbouring plant species and maintaining plant diversity of lowland landscapes dominated by agricultural areas? We compared historical maps and recent habitat maps to understand the effects of landscape changes on the vegetation pattern of the study region, in southern Hungary. Dikes and primary vegetation of the landscape were selected for intensive vegetation sampling. We compared the floristic similarity and the Shannon diversity of the vegetation types. We used ordinations to visualize relationships among the vegetation types and among dike vegetation and environmental variables. Our results indicated that profound changes have been brought about in the vegetation during the last 150 years, resulting in a transition from marshland to agricultural land. The species composition and pattern of dike vegetation strongly depended on their relative position to the river and their aspect. We conclude that dikes can harbour many vascular plants that are absent or rare in the surrounding habitats and therefore play a decisive role in maintaining plant diversity in agricultural landscapes.

Keywords: biodiversity loss; dike vegetation; grasslands; landscape change; Maros River; wetlands

1. Introduction

The last few centuries have witnessed considerable landscape changes in Europe, as technological advances have made agriculture more intensive and a high proportion of remaining areas have gradually been converted into arable fields, built-up areas, pastures and secondary forests (Bastian and Bernhardt 1993; Biró et al. 2008). Landscape changes become especially striking when the extent of historical and present wetland habitats are compared (Timmermann et al. 2006). In Europe, there was a boom in river regulation activities in the 19th century (Maltby and Blackwell 2005), which altered the natural flood pulse of many rivers and created new fields for agriculture. Due to these direct and indirect effects, riverine habitats have been destroyed (Varga et al. 2013).

In Western Europe, landscapes experienced severe changes in land use in the 20th century as well (Polus et al. 2007). Since World War II (1939–1945), agriculture has become increasingly intensified and the methods of agriculture have also changed profoundly (Erhardt 1985). In many Central and Eastern European countries, landscape changes were directly affected by the collectivisation and the extremely intensive agricultural exploitation during the communist era (e.g. Baessler and Klotz 2006, for East Germany; Feranec et al. 2007, for Slovakia). After the breakdown of the communist regimes (1989–1992), the intensity of agricultural use decreased in many regions (Kuemmerle et al. 2009). However, by that time, many semi-natural grassland patches had disappeared or changed markedly both in lowlands and highlands (Kamp et al. 2011; Sudnik-Wójcikowska et al. 2011). Today, semi-natural grasslands and their unique communities are among the most vulnerable ecosystems all over the globe (Dengler et al. 2014).

Many researchers have shown that remnants of native vegetation can survive in secondary and man-made habitats. Investigations were carried out e.g. on walls (Daniel and

Lecamp 2004), kurgans (burial mounds, also known as the ‘pyramids of the lowlands’) (Sudnik-Wójcikowska et al. 2011), river dikes (Liebrand and Sykora 1996), temporary pools on arable fields (Lukács et al. 2013; Takács et al. 2013) and in crop fields and edges (Fried et al. 2009; Kovács-Hostyánszki et al. 2011). Many studies showed that species richness and species diversity are higher in semi-natural habitats than in secondary ones (Barthlott et al. 2001; Acebey et al. 2003). In contrast, some have reported similarly high or even higher species richness and diversity in secondary habitats than in primary ones (van Andel 2001; Holz and Gradstein 2005).

Species composition of floodplains is strongly influenced not only by dispersal and disturbance processes, soil moisture and water level fluctuations but also by the presence or absence of dikes as well (Kingsford 2000; van Looy et al. 2003; Leyer 2004; Reinecke et al. 2015). River dikes are among the most extensive secondary habitats in the Hungarian lowland landscape with a total length of about 4.200 km (Felkai 2006). Before regulation, about 25% of the Hungarian lowland areas were flooded periodically. There were around 19,000 km^2 of functioning wetlands along the Tisza River (the second largest river in the Carpathian Basin) and its tributaries (e.g. the Maros River) in the 18^{th} century, which have decreased dramatically to around 530 km^2 after the regulations (Oláh and Oláh, 1996; Tockner and Stanford 2002). The river regulation works and the construction of dikes resulted in profound changes to river structure and function alike. Most river dikes were sown using different seed mixtures in order to reduce the impact of erosion and to produce fodder for livestock (Felkai 2006; Hoffmans et al. 2008). Besides these, river dikes can also be important from a conservation point of view, because they can support semi-natural habitats as well as endangered and protected plants and animals (Liebrand and Sykora 1996).

In this study, we assessed the conservation value of the river dikes along the Maros River (southern Hungary) in relation to the remnants of primarily herbaceous vegetation patches of this landscape. In order to understand the current species composition and vegetation characteristics, we also compared historical maps with recent vegetation maps of the area. Our specific objectives were to (1) examine how floodplain vegetation changed over the past 150 years; (2) compare the species composition and diversity patterns of the river dike vegetation with those of nearby primary vegetation types and (3) assess the role of river dikes in maintaining plant diversity in a lowland landscape dominated by agricultural areas.

2. Material and methods

2.1. Study site

The Maros River flows in a westerly direction and is one of the major rivers of the Carpathian Basin with a length of approximately 750 km and a catchment area of 30,000 km^2. The river originates from the Eastern Carpathian Mountains, Romania. Only the lowest 28 km are situated entirely in Hungary, while a 22 km long section forms the border between Hungary and Romania. The hydrograph for the Maros River can usually be characterized by two floods; snowmelt-induced floods occur in early spring, and rain-induced floods in early summer. Floods usually last for only a short time (6 days in average) (Kiss and Sipos 2007). On the Hungarian side of the river valley, the annual mean temperature is 10.5–10.6 °C and the average annual rainfall is 570 mm (Dövényi 2010). Typical soils along the Maros River valley are mostly alluvial protosoils and alluvial soils, but chernozems and alkali soils also occur (Jakab 1995).

Different types of deciduous forests (e.g. riverine willow-poplar forests, American poplar and other deciduous plantations) dominate the periodically inundated area. Only some semi-natural vegetation patches (e.g. closed steppes on loess and salt meadows) can be found in the never flooded, cultivated landscape (Margóczi et al. 2002). The Maros River and its current floodplain are included in the Natura 2000 network; some sections of the floodplain are part of the Körös–Maros National Park. River dikes along the Maros River have been reinforced after an extreme flood in 1970; therefore the age of the investigated grasslands is about 40 years. We defined primary habitats as habitats that have probably not been tilled in the last 100 years.

2.2. Data collection

Fieldwork was carried out on the Hungarian side, along the lowest 50 km stretch of the Maros River between 2010 and 2015 (Fig. 1). Habitat mapping was done by field surveys and aerial photo interpretation in four representative areas along the river (1: N46° 14’ E20° 14’; 2: N46° 14’ E20° 17’; 3: N46° 11’ E20° 29’; 4: N46° 8’ E20° 38’) in order to assess the current state of landscape vegetation (Fig. 1). The size of the selected areas was between 7 and 9 km^2, considering the location of the state border. Historical vegetation of the same areas was assessed by interpreting the map of the Second Military Survey (1864) of the Habsburg Empire. Habitats were identified using the National Habitat Classification System (Á-NÉR 2007) (Bölöni et al. 2011).

The herbaceous vegetation of the riverside and landside slopes of the northern and southern dikes of the Maros River was sampled using randomly arranged plots of 2 m × 2 m. The plots were placed on the upper two-thirds of dikes which are less influenced by the effects of floods. It also means that the species composition of these vegetation types remained relatively stable over the study period. In order to obtain representative samples, both the southern and northern dikes were divided into four sections (Fig.1). Five plots were taken from each section, therefore 20 plots were obtained for each dike slope ((LSD: landside slope of the southern dike (mostly south-facing); RSD: riverside slope of the southern dike (mostly north-facing); RND: riverside slope of the northern dike (mostly south-facing); LND: landside slope of the northern dike (mostly north-facing)) (Fig. 1, 2). For comparison, 20 plots of 2 m × 2 m were selected in each primary herbaceous vegetation type within a distance of up to 5 km from the Maros River. These vegetation types included mesotrophic wet meadows, closed steppes on loess, *Artemisia* salt steppes, salt meadows (together with transitional stands of salt meadows and *Achillea* steppes), and salt marshes. All habitats except mesotrophic wet meadows occur in the flood-protected area. As an initial step before sampling, we determined each patch size using aerial photographs and satellite images in combination with detailed field observations. The number of plots per patch ranged from one to five, depending on the patch sizes of each vegetation type. The sampled patches were selected randomly. The percentage cover of every vascular herb and tree sapling was estimated in May or early June in all 180 plots.

To understand the effect of dike position and slope exposure on species composition, soil samples were collected in three plots of 2 m × 2 m of each dike slope from the upper 20 cm of soil. Sampling sites were placed at least 1 km apart. Soil moisture (%) was determined gravimetrically, while soil organic matter content (%) was measured with spectrophotometer after wet oxidation by potassium dichromate and sulphuric acid. We also measured air temperature (°C) and air humidity (%) with wireless sensors for 24 hours 5 cm above the ground surface in these 12 plots. In addition, the presence/absence data of every vascular herb and tree sapling was estimated in the plots. Measurements were carried out in June 2014, after a dry period and under clear weather conditions. The names of plant species followed Király (2009).

2.3. Data analysis

We prepared habitat maps and determined the percentage of different vegetation types using ArcView GIS 3.2 (ESRI). For better understanding of vegetation changes, we used not only individual but also combined Á-NÉR habitat categories on the habitat maps. All vascular plant species recorded in the plots were classified according to their coenological preferences (Borhidi 1993). The proportion of the coenological groups within vegetation types was calculated using presence/absence data. The floristic similarity between the floras of the vegetation types (using the species lists of the plots) was calculated with Jaccard similarity index.

The diagnostic value of each species for each vegetation type was calculated, using the phi (Φ) coefficient of association (Chytrý et al. 2002). Species with $\Phi > 0.3$ were considered as diagnostic for individual vegetation types (Fisher's exact test, $p < 0.05$). In cases when a species appeared to be diagnostic for more than one vegetation type, only the vegetation type with the higher phi value was considered. Calculations were done with the JUICE 7.0.25 program (Tichý 2002).

Shannon diversity was calculated for each plot. After testing normality, a Kruskal-Wallis nonparametric test was performed to characterize the differences in diversity. Post hoc comparisons were carried out with Bonferroni-corrected Mann–Whitney U tests. Non-metric multidimensional scaling (NMDS) ordinations, based on square root transformed cover data and Bray-Curtis dissimilarity, were applied to compare the vegetation of the dikes to the primary vegetation of the landscape. For the calculations we used the ‘vegan’ community ecology package (Oksanen et al. 2015). At first, we performed an ordination based on all plots. For a better visualization of the differences among the plots of non-alkali vegetation and dikes, a second NMDS was performed, excluding the data of alkali vegetation. To analyse the relationships between the environmental variables and the species composition (presence/absence data) of different dike slopes, another NMDS ordination was performed by fitting environmental vectors onto the ordination space using the *envfit* function. We used the Jaccard index as distance measure. To evaluate the ordination, correlations between ordination values and fitted vectors were calculated. Linearity of fitted smooth surfaces on the ordination was assessed with generalized additive models (GAM) using the *ordisurf* function. All statistical analyses were performed in R statistical environment (R Development Core Team 2015).

Figures were prepared with Microsoft Excel and Adobe Photoshop CS2.

3. Results

3.1. Landscape change

According to the interpretation of the map of the Second Military Survey (1864) for the four areas, the landscape was mainly characterized by meadows (57%) at that time (Fig. 3). The amount of forests (17.5%), arable fields (16.4%), lakes and rivers (8.7%) and settlements (0.4%) was much lower. Nowadays, both the periodically flooded area and the former floodplain are characterized mainly by degraded habitats. According to the results of the vegetation mapping, the habitat structure of the Maros River valley is made up of the following: arable fields (41.3%), non-native deciduous tree rows and plantations (16.2%), secondary hardwood forests and shrubs (13.5%), riverine willow-poplar groves (9.4%), lakes and rivers (6.2%), mesotrophic wet meadows (4.8%), settlements (4.2%), secondary grasslands and tall herb communities (including the vegetation of dikes as well) (4%) and salt meadows (0.4%) (Fig. 3). When using the same habitat categories as those in the interpretation of the map of the Second Military Survey (1864), the results were as follows: meadows (9.2%), forests (39.1%), arable fields (41.3%), lakes and rivers (6.2%) and settlements (4.2%). Comparing the changes between the two periods, the proportion of arable fields and forests increased considerably (from 16.4% to 41.3 % and from 17.5% to 39.1%, respectively), whereas the proportion of grasslands and meadows decreased drastically (from 57% to 9.2%).

3.2. Species composition and Shannon diversity

In total, 231 vascular plant species were found in the 180 plots. The highest number of species was found on LND (90), while the lowest number of species in salt marshes (47) (Table 1). Of the 231 species, 6 were restricted to LSD, 13 to RSD, 4 to RND, 8 to LND, 5 to mesotrophic wet meadows, 13 to closed steppes on loess, 8 to *Artemisia* salt steppes, 11 to salt meadows and 20 to salt marshes, thus 38% of the species were found in only one vegetation type.

Jaccard similarity was low (< 25% in all cases) between dike vegetation and alkali vegetation. However, floristic similarity was much greater between LSD and RND (54%), RSD and RND (42%), RSD and LND (44%), LSD and LND (46%), RND and LND (60%), RSD and mesotrophic wet meadows (44%), LND and closed steppes on loess (43%) and salt meadows and *Artemisia* salt steppes (45%) (Table 1). Generally, similarity between dike vegetation types was higher than between dike vegetation types and the primary vegetation types.

The list of diagnostic species for each vegetation type is given in Table 2. The number of diagnostic species was the highest on RSD (24) and the lowest on LND (3). The proportion of semi-dry grassland species (Sedo-Scleranthetea and Festuco-Brometea) (e.g. *Festuca rupicola* Heuff. and *Salvia austriaca* Jacq.) was rather high on LSD (23%), LND (20.1%) and in closed steppes on loess (27%), and relatively low on RSD (9.5%) and in salt meadows (5.1%) and salt marshes (0.5%). Marshland species (Molinio-Arrhenatheretea) (e.g. *Alopecurus pratensis* L. and *Poa pratensis* L. s.l.) played an important role in structuring salt meadows (12.6%), salt marshes (15.1%) and the vegetation of RSD (12.8%). The proportion of wetland species (Lemnetea and Phragmitetea) (e.g. *Lemna minor* L. and *Glyceria fluitans* (L.) R. Br.) was especially high in salt marshes (20.5%). Alkali species (Festuco-Puccinellietea) (e.g. *Beckmannia eruciformis* (L.) Host and *Trifolium angulatum* Waldst. et Kit.) played an important role only in *Artemisia* salt steppes (34.6%), salt meadows (34.5%)

and salt marshes (8.1%). The proportion of the species of disturbed habitats (Agropyretea, Agrostietea stoloniferae, Artemisietea, Bidentetea, Chenopodietea and Secalietea) (e.g. *Allium atropurpureum* Waldst. et Kit. and *Calepina irregularis* (Asso) Thell.) ranged from 9.2% (in salt marshes) to 30.1% (on RND). The rate of plant invasion was very low in all vegetation types.

Habitat type had a significant effect on Shannon diversity (X^2 = 109.9, p < 0.001). The Mann–Whitney tests showed that diversity on RSD, RND and LND and in mesotrophic wet meadows and closed steppes on loess was significantly higher (p < 0.005) than in salt meadows and salt marshes (Fig. 4). However, their diversities were not significantly different from each other. This was the same for LSD and LND. The vegetation of LSD, LND and *Artemisia* salt steppes was more diverse (p < 0.005) than that of salt marshes.

3.3. Vegetation pattern and vegetation-environment relationships

The NMDS ordination of all plots (stress: 0.24) revealed the differences between the alkali and non-alkali vegetation types of the landscape (Fig. 5a). Axis 1 opposed the plots of non-alkali vegetation (mesotrophic wet meadows, closed steppes on loess and dike vegetation), on the left, and plots of alkali vegetation (*Artemisia* salt steppes, salt meadows and salt marshes), on the right. According to the second NMDS (stress: 0.19) (Fig. 5b), in which only non-alkali vegetation data were used, the plots of LSD and the plots of RSD were strongly separated from each other. The vegetation of RSD was rather similar to mesotrophic wet meadows, while most plots of LND were rather similar to those of closed steppes on loess. Many plots of LSD were well separated on axis 1. Among the plots of RND, there was considerable variation, whereas the plots of mesotrophic wet meadows were the most similar to one another.

The NMDS of presence/absence data (stress: 0.06) revealed a similar pattern of dike vegetation as observed with the NMDS of the non-alkali vegetation data (Fig. 6). Axis 1 separated dike vegetation along a humidity (r = 0.93; p < 0.001), moisture (r = 0.88; p < 0.001) and temperature (r = -0.82; p < 0.005) gradients and axis 2 along a soil organic matter (r = 0.65; p < 0.05) gradient. The warmest and driest conditions were found on LSD, while RSD was the coldest and most humid. Soil organic matter content was the highest on RND. Environmental conditions of RND and LND were more similar to one another than those of LSD and RSD. The difference between LSD and RSD in soil organic matter content was rather small.

4. Discussion

Studying the driving forces of landscape change has a long tradition in geography and landscape research (Hersperger and Bürgi 2009) and has received increased interest in ecology and vegetation science (Biró et al. 2013). Similar to some other Central European countries, the dramatic change in vegetation structure and the loss of natural habitats of the Maros River valley can be associated to the landscape changes of the last two centuries. Comparing historical maps and our current habitat maps, we found that the present vegetation pattern is significantly influenced by river regulation activities and agricultural developments, which were the main components of landscape change all over Europe over the past few centuries (Zomeni et al. 2008; Varga et al. 2013). The most prominent change in vegetation distribution of the Maros River valley was the decrease of land cover types dominated by flood-meadow and marshland species.

Compared with the primary vegetation types of the landscape, both species richness and Shannon diversity were relatively high on the dikes of the Maros River. Current diversity patterns may be traced back to several factors. Most of the river dikes were sown using different seed mixtures after the reconstruction, which determined the initial species composition and species richness. In Hungary, the composition of the sown seed mixtures changed during the last few centuries in line with the temporal changes in mass propagation trends and in the species composition of commercially available seed mixtures (Felkai 2006). Nowadays, most commercially available seed mixtures are often non-native cultivars with foreign origin (mostly from the Netherlands and Denmark) (Török et al. 2011a). The occurrence of some species on the dikes is certainly related to the sown seed mixtures (e.g. *Bromus erectus* Huds. on the dikes of the Maros River). Floodplains and rivers are ecological corridors that promote the dispersal of plant propagules and connect habitats (Gallé et al. 1995;

Ward et al. 2002). Large numbers of plant propagules can be transported by rivers over large distances and deposited on the riverside slopes of the dikes and riparian zones (Johansson et al. 1996; Jansson et al. 2000). Thus, under the different stages of succession of dike vegetation, many dicot species can also colonize the dike slopes from the landscape species pool. A recent study by Rooney et al. (2013) showed that the effects of intensive flooding can supersede the effects of water and sediment quality on the floating plant communities of highly connected wetlands. Because both flood intensity, water quality and the rate of connection affect the number and types of transported propagules as well (Jansson et al. 2005), these factors may significantly determine the direction of succession on the riverside slopes of dikes. Also, similar to other habitats where vegetation composition is largely defined by microclimatic heterogeneity (Bátori et al. 2014), the ecological conditions (e.g. air temperature and soil moisture) of the differently oriented dike slopes are highly variable. In the Northern Hemisphere, south-facing slopes receive more solar radiation, and thus are usually warmer and drier than north-facing ones (Erdős et al. 2012). However, as we can see in Fig. 6, soil moisture was also relatively high on the south-facing riverside slope, resulting in a special mixture of marshland and dry grassland species and high species diversity. In addition, the regular mowing of grasslands can enhance biodiversity (Collins et al. 1998; Valkó et al. 2012). Secondary grasslands of the dikes along the Maros River are mown (two or three times a year), thus mowing may also be an important factor contributing to the relatively high species diversity. Only a small section of dikes is grazed.

Many researchers suggest that secondary habitats may play an important role in the preservation of species diversity in landscapes where agricultural activities are the dominant land use practice. For example, orchards may act as refuges for spiders (Bogya et al. 1999), planted shade coffee plantations for birds (Greenberg et al. 1997), walls for ferns (Láníková and Lososová 2009), high-way stormwater ponds for aquatic macroinvertebrates (Le Viol et al. 2009) and graveyards for orchids (Löki et al. 2015). Line habitats, such as hedgerows, river dikes and road verges, have an especially important role, because they contribute to the recruitment or reestablishment of populations via migration from other habitats (Corbit et al. 1999; Bellamy et al. 2000). Torma and Császár (2013) found numerous true bug species on the dikes of the Tisza River (Hungary) and concluded that these habitats have a great importance for conservation of insect diversity in agricultural landscapes. The same conclusion has been reached by Gallé et al. (2011) when studying the spider assemblages of the floodplain areas of southern Hungary. River dikes of the southern part of the Upper Rhine valley have been identified as an important habitat for the endangered flightless beetle *Dorcadion fuliginator* L. (Baur et al. 2002). Several authors noted that the age of habitats may strongly influence the species composition and diversity patterns of landscapes (van Adel 2001; Renner et al. 2006). Older secondary habitats might be more suitable for the preservation of biodiversity and a high degree of species richness than younger ones (Chazdon et al. 2009). The maintenance of both older secondary habitats and remnants of primary habitats is essential to many organisms within highly-fragmented agricultural landscapes.

River regulation resulted in profound changes along the Maros River and the landscape changed from a marshland-grassland complex to a primarily agricultural landscape. It has been stressed that conservation efforts must prioritize the preservation of the remnant pristine and threatened habitat patches and should channel the efforts to increase the population size of endangered plant species by targeted restoration (planting or hay transfer from pristine habitats, Kirmer et al. 2008; Lencová and Prach 2011; Török et al. 2011a). Establishment of new populations by sowing of seeds or planting individuals deriving from much larger populations or from *ex situ* populations provides an opportunity to avoid their extinction (Hamilton 1994). In southern Hungary, there are many examples where endangered loess and dry grassland species have been successfully reintroduced into pristine habitats (e.g. loess grassland patches), or into some degraded habitats (recovering grasslands in former cropfields). Most of the loess areas have been ploughed in Hungary, thus reintroduction of species and habitat restoration has a crucial role in maintaining endangered plant species (Török et al. 2011b). Today, many populations of these species occur only in road verges and on railway embankments in southern Hungary (Király 2009). Our results revealed that the vegetation of the LND along the Maros River was rather similar to that of the loess grassland fragments in the landscape. Thus this slope of the dike system, though a secondary habitat, can be considered a refuge for many species and a potential target site for the passive as well as actively assisted colonization of valuable loess and dry grassland species in the future. This is especially true for the dike sections between Makó and Nagylak (Fig. 1), where we have already found some rare loess and dry grassland species (e.g. *Agropyron cristatum* (L.) Gaertn., *Allium atropurpureum*, *Allium rotundum* L. subsp. *rotundum*, *Carthamus lanatus* L., *Centaurea scabiosa* L. s.l., *Ornithogalum brevistylum* Wolfner and *Thalictrum minus* L.) (see Fig. 2). However, because the primary role of dikes is the protection against floods, conservationists must work with other stakeholders (e.g. water managers and local people) to avoid conflicts and to improve the success of conservation programs.

Flood risk and vulnerability are likely to have grown in many European areas due to land-use and land-cover changes and to climate change (Mudelsee et al. 2003; Dankers and Feyen 2008). Climate models have identified significant changes in the magnitude and frequency of precipitation for the catchment area of the large rivers in Central Europe (Kundzewicz et al. 2005). As floods are substantial natural hazards, to preserve and improve the vegetation of species-rich grasslands on river dikes, we need to take into account the possible effects of dike reconstructions. If during dike reconstructions (e.g. after damages caused by massive floods) strips of the original species-rich vegetation are kept unaffected or the upper soil layer can be put aside as complete sods and be replaced as the new topsoil after the reconstruction, they can function as sources of propagules and contribute to the rapid redevelopment of dike vegetation (Liebrand and Sykora 1996).

Secondary habitats like river dikes may play a decisive role in maintaining plant diversity in highly fragmented agricultural landscapes. However, further investigations are necessary to expand our understanding of the relationship between river water level fluctuations, management strategies and vegetation pattern changes on the dike slopes in order to make better predictions for erosion protection of soils and for nature conservation activities during climate change.

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6. References

Acebey A, Gradstein, SR, Krömer T (2003) Species richness and habitat diversification of bryophytes in submontane rain forest and fallows of Bolivia. Journal of Tropical Ecology 19: 9–18

Baessler C, Klotz S (2006) Effects of changes in agricultural land-use on landscape structure and arable weed vegetation over the last 50 years. Agriculture, Ecosystems & Environment 115: 43–50

Barthlott W, Schmit-Neuerburg V, Nieder J, Engwald, S (2001) Diversity and abundance of vascular epiphytes: a comparison of secondary vegetation and primary montane rain forest in the Venezuelan Andes. Plant Ecology 152: 145–156

Bastian O, Bernhardt A (1993) Anthropogenic landscape changes in Central Europe and the role of bioindication. Landscape Ecology 8: 139–151

Baur B, Zschokke S, Coray A, Schläpfer M, Erhardt A (2002) Habitat characteristics of the endangered flightless beetle *Dorcadion fuliginator* (Coleoptera: Cerambycidae): implications for conservation. Biological Conservation 105: 133–142

Bátori Z, Csiky J, Farkas T, E Vojtkó A, Erdős L, Kovács D, Wirth T, Körmöczi L, Vojtkó A (2014) The conservation value of karst dolines for vascular plants in woodland habitats of Hungary: refugia and climate change. International Journal of Speleology 43: 15–26

Bellamy PE, Shore RF, Ardeshir D, Treweek JR, Sparks TH (2000) Road verges as habitat for small mammals in Britain. Mammal Review 30: 131–139

Biró M, Czúcz B, Horváth F, Révész A, Csatári B, Molnár Zs (2013) Drivers of grassland loss in Hungary during the post-socialist transformation (1987-1999). Landscape Ecology 28: 789–803

Biró M, Révész A, Molnár Zs, Horváth F, Czúcz B (2008) Regional habitat pattern of the Danube-Tisza Interfluve in Hungary II. Acta Botanica Hungarica 50: 19–60

Bogya S, Szinetár Cs, Markó V (1999) Species composition of spider (Araneae) assemblages in apple and pear orchards in the Carpathian Basin. Acta Phytopathologica et Entomologica Hungarica 34: 99–121

Bölöni J, Molnár Zs, Kun, A (eds) (2011) Magyarország élőhelyei. Vegetációtípusok leírása és határozója. MTA Ökológiai és Botanikai Kutatóintézete, Vácrátót

Borhidi A (1993) Social behaviour types of the Hungarian flora, its naturalness and relative ecological indicator values. Pécs, Hungary

Chazdon RL, Peres CA, Dent D, Sheil D, Lugo AE, Lamb D, Stork NE, Miller S (2009) The potential for species conservation in tropical secondary forests. Conservation Biology 23: 1406–1417

Chytrý M, Tichý L, Holt J, Botta-Dukát Z (2002) Determination of diagnostic species with statistical fidelity measures. Journal of Vegetation Science 13: 79–90

Collins SL, Knapp AK, Briggs JM, Blair JM, Steinauer EM (1998) Modulation of diversity by grazing and mowing in native tallgrass prairie. Science 280: 745–747

Corbit M, Marks PL, Gardescu S (1999) Hedgerows as habitat corridors for forest herbs in central New York, USA. Journal of Ecology 87: 220–232

Daniel H, Lecamp E (2004) Distribution of three indigenous fern species along a rural-urban gradient in the city of Angers, France. Urban Forestry & Urban Greening 3: 19–27

Dankers R, Feyen L (2008) Climate change impact on flood hazard in Europe: An assessment based on high-resolution climate simulations. Journal of Geophysical Research 113. doi: 101029/2007JD009719

Dengler J, Janisová M, Török P, Wellstein C (2014) Biodiversity of Palaearctic grasslands: a synthesis. Agriculture, Ecosystems & Environment 182: 1–14

Dövényi Z (ed) (2010) Magyarország kistájainak katasztere. MTA Földrajztudományi Kutatóintézet, Budapest

Erdős L, Méri Á, Bátori Z, Gallé R, Körmöczi L (2012) North-south facing vegetation gradients in the Villány mountains: a case study on the population and the community level. Pakistan Journal of Botany 44: 927–932.

Erhardt A (1985) Diurnal Lepidoptera: sensitive indicators of cultivated and abandoned grassland. Journal of Applied Ecology 22: 849–861

Felkai BO (2006) Gyepborítású árvízvédelmi földgátak ökonómiai kérdései. Dissertation, Szent István University

Feranec J, Hazeu G, Christensen S, Jaffrain, G (2007) Corine land cover change detection in Europe (case studies of the Netherlands and Slovakia). Land Use and Policy 24: 234–247

Fried G, Petit S, Dessaint F, Reboud X (2009) Arable weed decline in Northern France: Crop edges as refugia for weed conservation? Biological Conservation 142: 238–243

Gallé L, Margóczi K, Kovács É, Györffy Gy, Körmöczi L, Németh L (1995) River valleys: Are they ecological corridors? Tiscia 29: 53–58

Gallé R, Vesztergom N, Somogyi T (2011) Environmental conditions affecting spiders in grasslands at the lower reach of the River Tisza in Hungary. Entomologica Fennica 22: 29–38

Greenberg R, Bichier P, Sterling J (1997) Bird populations in rustic and planted shade coffee plantations of Eastern Chiapas, México. Biotropica 29: 501–514

Hamilton MB (1994) Ex situ conservation of wild plant species: time to reassess the genetic assumptions and implications of seed banks. Conservation Biology 8: 39–49

Hersperger AM, Bürgi, M (2009) Going beyond landscape change description: Quantifying the importance of driving forces of landscape change in a Central Europe case study. Land Use and Policy 26: 640–648

Hoffmans G, Akkerman GJ, Verheij H, van Hoven A, van der Meer J (2008) The erodibility of grassed inner dike slopes against wave overtopping. ASCE, Proc. ICCE 2008, Hamburg: 3224–3236

Holz I, Gradstein RS (2005) Cryptogamic epiphytes in primary and recovering upper montane oak forests of Costa Rica – species richness, community composition and ecology. Plant Ecology 178: 89–109

Jakab S (1995) Soils of the flood plain of the Mures (Maros) River. Tiscia Monograph Series 1: 25–46

Jansson R, Nilsson C, Renofalt B (2000) Fragmentation of riparian floras in rivers with multiple dams. Ecology 81: 899–903

Jansson R, Zinko U, Merritt DM, Nilsson C (2005) Hydrochory increases riparian plant species richness: a comparison between a free-flowing and a regulated river. Journal of Ecology 93: 1094–1103.

Johansson ME, Nilsson C, Nilsson E (1996) Do rivers function as corridors for plant dispersal? Journal of Vegetation Science 7: 593–598

Kamp J, Urazaliev R, Donald PF, Hölzel N (2011) Post-Soviet agricultural change predicts future declines after recent recovery in Eurasian steppe bird populations. Biological Conservation 144: 2607–2614

Kingsford RT (2000) Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. Austral Ecology 25: 109–127.

Király G (ed) (2009) Új magyar füvészkönyv. Aggteleki Nemzeti Park Igazgatóság, Jósvafő

Kirmer A, Tischew S, Ozinga WA, von Lampe M, Baasch A, Groenendael JM (2008) Importance of regional species pools and functional traits in colonization processes: predicting re-colonization after large-scale destruction of ecosystems. Journal of Applied Ecology 45: 1523–1530

Kiss T, Sipos Gy (2007) Braid-scale channel geometry changes in a sand-bedded river: Significance of low stages. Geomorphology 84: 209–221

Kovács-Hostyánszki A, Batáry P, Báldi A, Harnos A (2011) Interaction of local and landscape features in the conservation of Hungarian arable weed diversity. Applied Vegetation Science 14: 40–48

Kuemmerle T, Müller D, Griffiths P, Rusu M (2009) Land use change in Southern Romania after the collapse of socialism. Regional Environmental Change 9: 1–12

Kundzewicz ZW, Ulbrich U, Brücher T, Graczyk D, Krüger A, Leckebusch GC, Menzel L, Pińskwar I, Radziejewski M, Szwed M (2005) Summer floods in Central Europe – Climate change track? Natural Hazards 36: 165–189

Láníková D, Lososová, Z (2009) Rocks and walls: natural versus secondary habitats. Folia Geobotanica 44: 263–280

Lencová K, Prach K (2011) Restoration of hay meadows on ex-arable land: commercial seed mixtures vs spontaneous succession. Grass and Forage Science 66: 265–271

Le Viol I, Mocq J, Julliard R, Kerbiriou C (2009) The contribution of motorway stormwater retention ponds to the biodiversity of aquatic macroinvertebrates. Biological Conservation 142: 3163–3171

Leyer I (2004) Effects of dykes on plant species composition in a large lowland river floodplain. River Research and Applications 20: 813–827

Liebrand CIJM, Sykora KV (1996) Restoration of semi-natural, species-rich grasslands on river dikes after reconstruction. Ecological Engineering 7: 315–326

Löki V, Tökölyi J, Süveges K, Lovas-Kiss Á, Hürkan K, Sramkó G, Molnár VA (2015) The orchid flora of Turkish graveyards: A comprehensive field survey. Willdenowia 45: 231–243

Lukács BA, Sramkó G, Molnár VA (2013) Plant diversity and conservation value of continental temporary pools. Biological Conservation 158: 393–400

Maltby E, Blackwell MSA (2005) Managing riverine environments in the context of new water policy in Europe. International Journal of River Basin Management 3: 133–141

Margóczi K, Drăgulescu C, Macalik K, Makra, O (2002) Vegetation description of representative habitat complexes along the Maros (Mures) river. Tiscia Monograph Series 6: 45–50

Mudelsee M, Börngen M, Tetzlaff G, Grünewald U (2003) No upward trends in the occurrence of extreme floods in central Europe. Nature 425: 166–169

Oksanen J, Blanchet FG, Kindt R, Legendre P, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens MHH, Wagner H (2015). vegan: Community Ecology Package. R package version 2.2-1. http://CRAN.R-project.org/package=vegan

Oláh J, Oláh M (1996) Improving landscape nitrogen metabolism in the Hungarian lowlands. Ambio 25: 331–335

Polus E, Vandewoestijne S, Choutt J, Baguette M (2007) Tracking the effects of one century of habitat loss and fragmentation on calcareous grassland butterfly communities. Biodiversity and Conservation 16: 3423–3436

R Development Core Team (2015) R: A language and environment for statistical computing R Foundation for Statistical Computing. http://wwwR-projectorg

Reinecke MK, Brown CA, Esler KJ, King JM, Kleynhans MT, Kidd M (2015) Links between lateral vegetation zones and river flow. Wetlands 35: 473–486.

Renner SC, Waltert M, Mühlenberg M (2006) Comparison of bird communities in primary vs. young secondary tropical montane cloud forest in Guatemala. Biodiversity and Conservation 15: 1545–1575

Rooney RC, Carli C, Bayley SE (2013) River connectivity affects submerged and floating aquatic vegetation in floodplain wetlands. Wetlands 33: 1165–1177.

Sudnik-Wójcikowska B, Moysiyenko II, Zachwatowicz M, Jabłońska E (2011) The value and need for protection of kurgan flora in the anthropogenic landscape of steppe zone in Ukraine. Plant Biosystems 145: 638–653

Takács A, Schmotzer A, Jakab G, Deli T, Mesterházy A, Király G, Lukács BA, Balázs B, Perić R, Eliáš P jun, Sramkó G, Tökölyi J, Molnár VA (2013) Key environmental variables affecting the distribution of Elatine hungarica in the Pannonian Basin. Preslia 85: 193–207

Tichý L 2002 JUICE, software for vegetation classification. Journal of Vegetation Science 13: 451–453

Timmermann T, Margóczi K, Takács G, Vegelin K (2006) Restoration of peat-forming vegetation by rewetting species-poor fen grasslands. Applied Vegetation Science 9: 241–250

Tockner K, Stanford JA (2002) Riverine flood plains: present state and future trends. Environmental Conservation 29: 308–330

Torma A, Császár P (2013) Species richness and composition patterns across trophic levels of true bugs (Heteroptera) in the agricultural landscape of the lower reach of the Tisza River Basin. Journal of Insect Conservation 17: 35–51

Török P, Kelemen A, Valkó O, Deák B, Lukács B, Tóthmérész B (2011b) Lucerne-dominated fields recover native grass diversity without intensive management actions. Journal of Applied Ecology 48: 257–264

Török P, Vida E, Deák B, Lengyel Sz, Tóthmérész B (2011a) Grassland restoration on former croplands in Europe: an assessment of applicability of techniques and costs. Biodiversity and Conservation 20: 2311–2332

Valkó O, Török P, Matus G, Tóthmérész B (2012) Is regular mowing the most appropriate and cost-effective management maintaining diversity and biomass of target forbs in mountain hay meadows? Flora 207: 303–309

van Andel T (2001) Floristic composition and diversity of mixed primary and secondary forests in northwest Guyana. Biodiversity and Conservation 10: 1645–1682

van Looy K, Honnay O, Bossuyt B, Hermy M (2003) The effects of river embankment and forest fragmentation on the plant species richness and composition of floodplain forests in the Meuse valley, Belgium. Belgian Journal of Botany 136: 97–108

Varga K, Dévai Gy, Tóthmérész B (2013) Land use history of a floodplain area during the last 200 years in the Upper-Tisza region (Hungary). Regional Environmental Change 13: 1109–1118

Ward JV, Mallard F, Tockner K (2002) Landscape ecology: a framework for integrating pattern and process in river corridors. Landscape Ecology 17: 35–45

Zomeni M, Tzanopoulos J, Pantis JD (2008) Historical analysis of landscape change using remote sensing techniques: An explanatory tool for agricultural transformation in Greek rural areas. Landscape and Urban Planning 86: 38–46

Figure caption:

Fig. 1 Location of the study sites along the Maros River in Hungary. The numbers (1-4) indicate the areas in which vegetation mapping was done. Arrows indicate the borders of dike sections. LSD: landside slope of the southern dike; RSD: riverside slope of the southern dike; RND: riverside slope of the northern dike; LND: landside slope of the northern dike

Fig. 2 Vegetation of the dike slopes (A–D) of the Maros River and some rare (E–H) and red-listed (E and H) grassland species found on them. A: landside slope of the southern dike (LSD); B: riverside slope of the southern dike (RSD); C: riverside slope of the northern dike (RND); D: landside slope of the northern dike (LND); E: *Allium atropurpureum*; F: *Ornithogalum brevistylum*; G: *Centaurea scabiosa* s.l. and H: *Carthamus lanatus*. Photos: Z. Bátori

Fig. 3 Historical (1a–4a) and current habitats (1b–4b) of four representative areas along the Maros River. A: arable fields; B: meadows (current: mesotrophic wet meadows); C: floodplain forests (current: riverine willow-poplar groves); D: settlements; E: lakes and rivers; F: secondary grasslands and tall herb communities (including the vegetation of dikes as well); G: secondary hardwood forests and shrubs; H: non-native deciduous tree rows and plantations and I: salt meadows

Fig. 4 Shannon diversity of the vegetation types ((A: landside slope of the southern dike (LSD); B: riverside slope of the southern dike (RSD); C: riverside slope of the northern dike (RND); D: landside slope of the northern dike (LND); E: mesotrophic wet meadows; F:

closed steppes on loess; G: *Artemisia* salt steppes; H: salt meadows; I: salt marshes)) of the Maros River valley. Boxes not sharing a letter (a-d) are significantly ($p < 0.05$) different

Fig. 5 NMDS ordination diagrams for the 180 plots of all investigated vegetation types (a) as well as for the 120 plots of non-alkali vegetation types (b). Stress values were 0.24 and 0.19, respectively. White circle: landside slope of the southern dike (LSD); black circle: riverside slope of the southern dike (RSD); cross: riverside slope of the northern dike (RND); grey square: landside slope of the northern dike (LND); white triangle: mesotrophic wet meadows; black line: closed steppes on loess; black diamond: *Artemisia* salt steppes; white diamond: salt meadows; grey triangle: salt marshes

Fig. 6 NMDS ordination diagrams of the dike vegetation, overlaid with smooth fitted surfaces of environmental variables (air temperature, soil organic matter, soil moisture and air humidity). Arrows indicate the main direction of gradients. The stress value was 0.06. White circle: landside slope of the southern dike (LSD); black circle: riverside slope of the southern dike (RSD); cross: riverside slope of the northern dike (RND); grey square: landside slope of the northern dike (LND)

Figures:

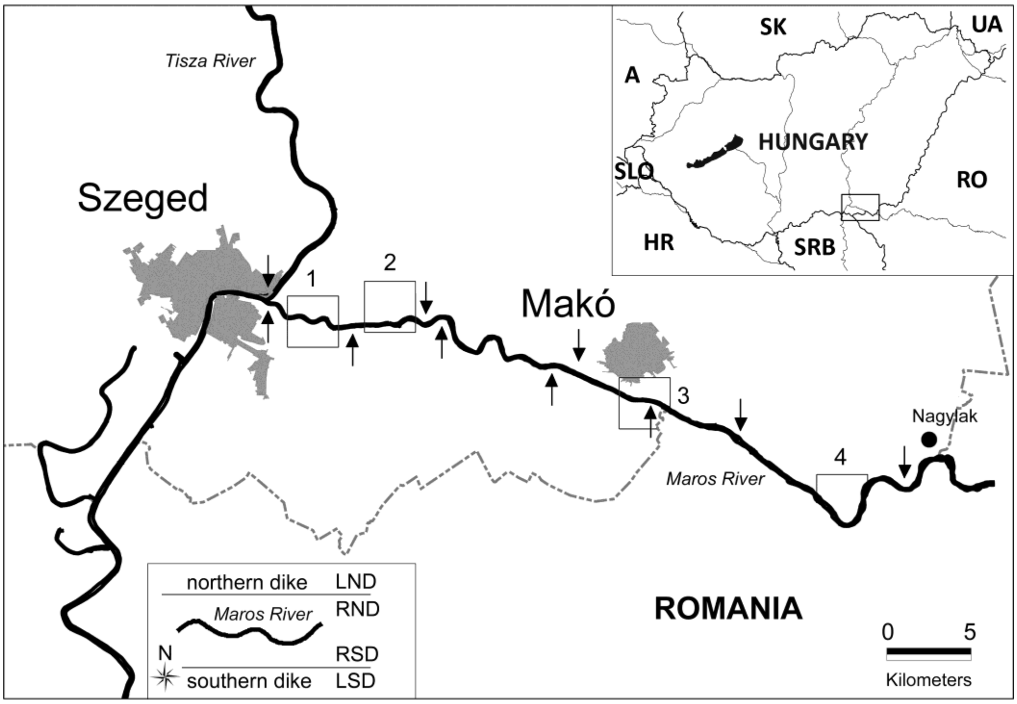
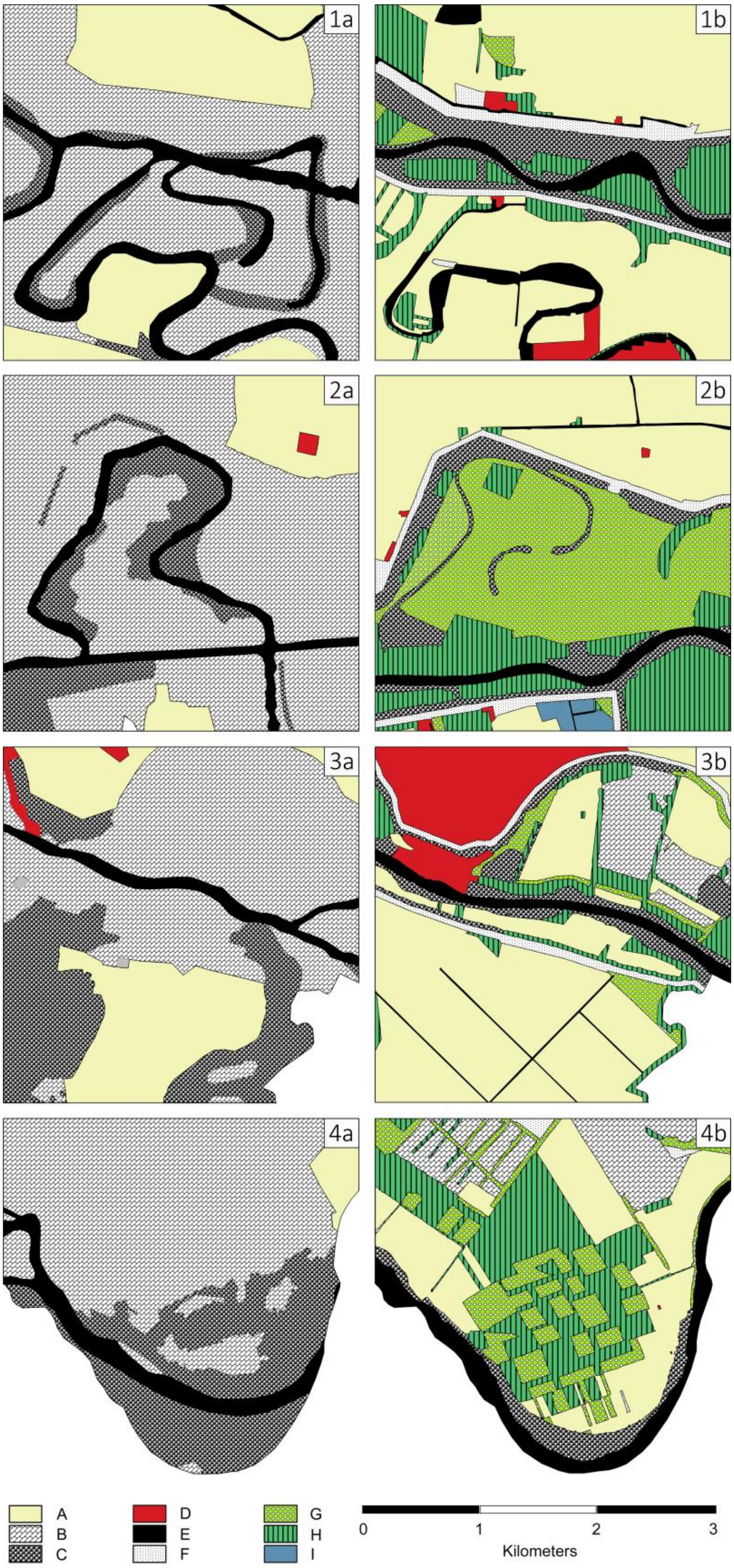


Fig. 1

Fig. 2



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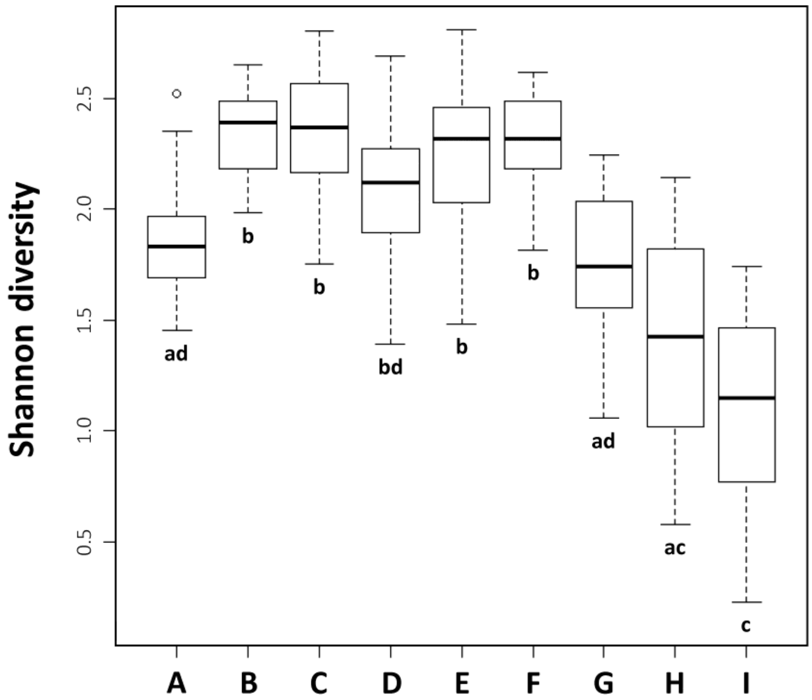

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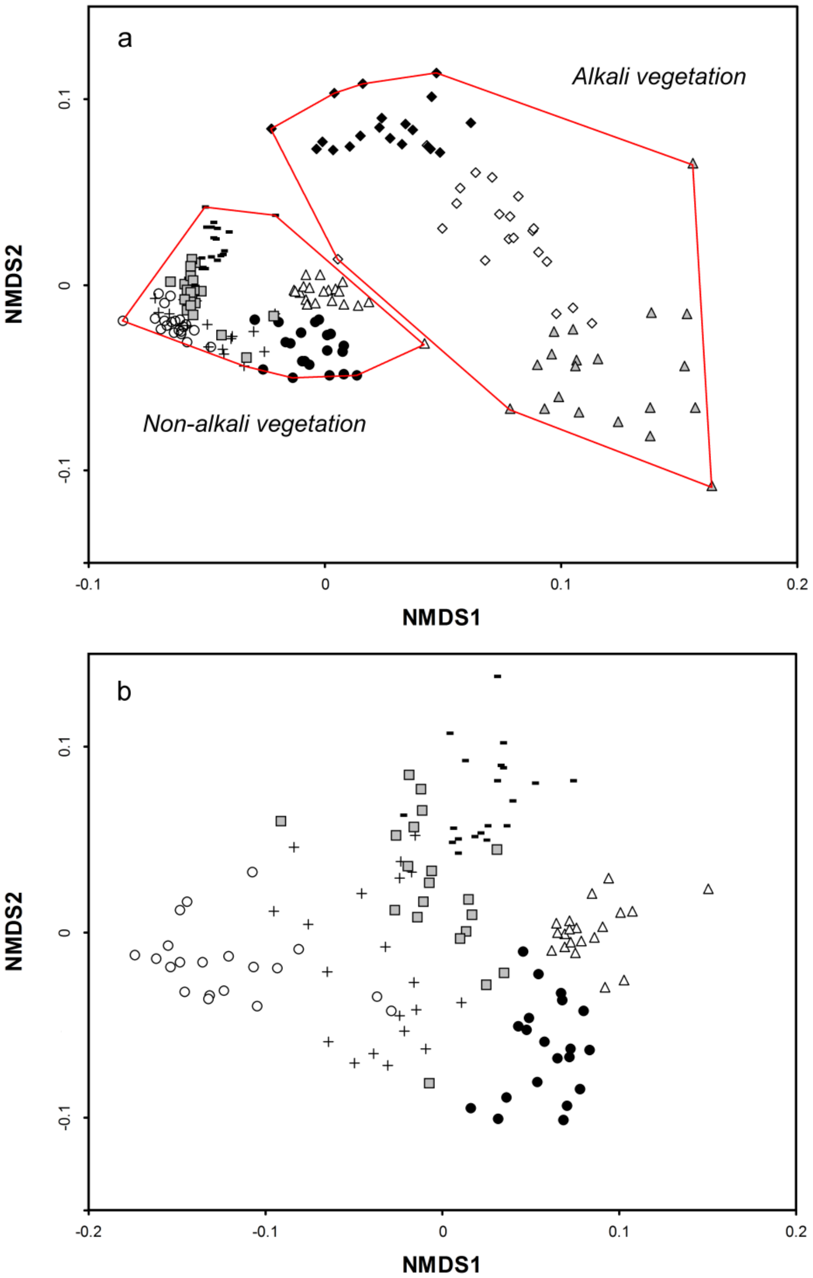
a
Alkali vegetation
Non-alkali vegetation
NMDS2
0.1
0
-0.1
-0.1
0
0.1
0.2
NMDS1
b
NMDS2
0.1
0
-0.1
-0.2
-0.1
0
0.1
0.2
NMDS1

Fig. 5

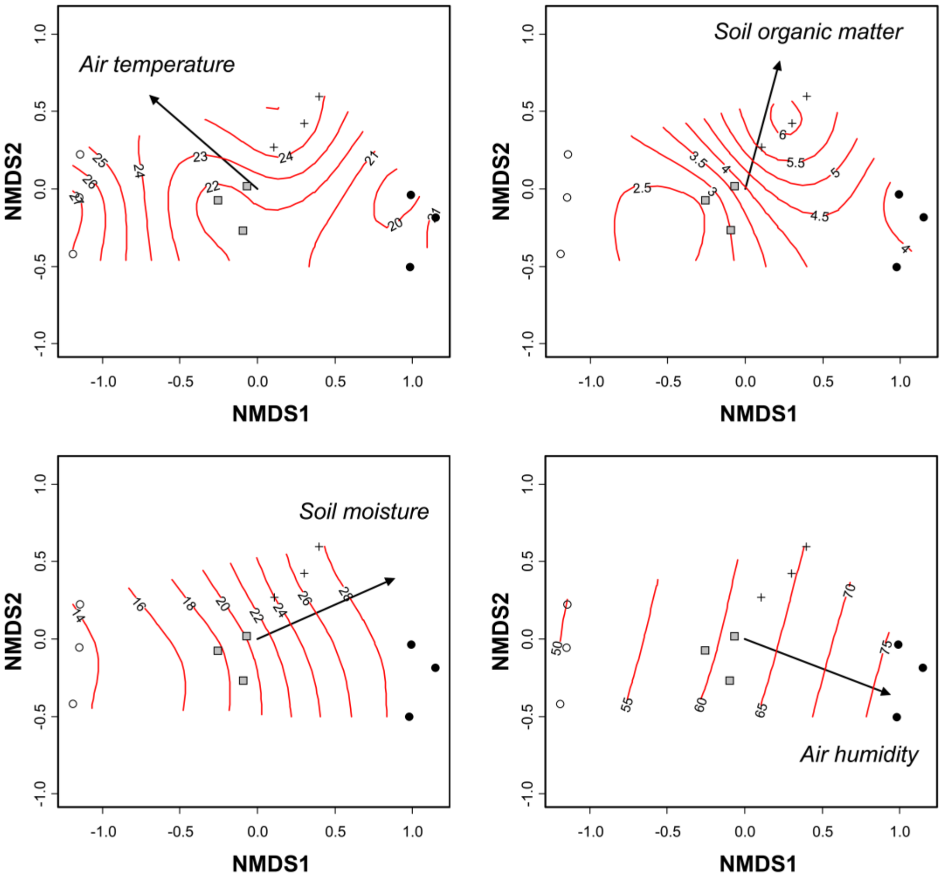
Air temperature
Soil organic matter
Soil moisture
Air humidity
NMDS1
NMDS2

Fig. 6

Tables:

Table 1 Species richness, similarity (Jaccard index), and number of shared species (in parenthesis) among the vegetation types (180 plots) of the Maros River valley. LSD: landside slope of the southern dike; RSD: riverside slope of the southern dike; RND: riverside slope of the northern dike; LND: landside slope of the northern dike. Similarity values above 0.40 are grey-shaded

	Species richness	LSD	RSD	RND	LND	Mesotrophic wet meadows	Closed steppes on loess	Artemisia salt steppes	Salt meadows	Salt marshes
LSD	62		(40)	(51)	(48)	(32)	(33)	(25)	(21)	(6)
RSD	84	0.38		(52)	(53)	(47)	(33)	(22)	(26)	(12)
RND	83	0.54	0.42		(65)	(42)	(43)	(26)	(26)	(6)
LND	90	0.46	0.44	0.60		(45)	(51)	(29)	(30)	(7)
Mesotrophic wet meadows	69	0.32	0.44	0.38	0.39		(35)	(26)	(32)	(13)
Closed steppes on loess	79	0.31	0.25	0.36	0.43	0.31		(38)	(34)	(6)
Artemisia salt steppes	66	0.24	0.17	0.21	0.23	0.24	0.36		(45)	(10)
Salt meadows	78	0.18	0.19	0.19	0.22	0.28	0.28	0.45		(22)
Salt marshes	47	0.06	0.10	0.05	0.05	0.13	0.05	0.10	0.21	

Table 2 Synoptic table of the vegetation of dikes (A-D) and of the primary herbaceous vegetation types of the Maros River valley (E-I). Within blocks of significant ($p < 0.05$) diagnostic species (values grey-shaded), species are ranked by decreasing fidelity. Species with $\Phi \times 100 < 30$ were not included in the groups of diagnostic species

	A	B	C	D	E	F	G	H	I
No. of relevés	20	20	20	20	20	20	20	20	20
A. Landside slope of the southern dike (LSD)									
Bromus tectorum	80.7								
Erodium cicutarium	76								
Alyssum alyssoides	72.5								
Calepina irregularis	58.6		46.6						
Poa bulbosa	57.4		16.5						
Medicago minima	51.6								
Bromus hordeaceus	41.8		27.5	31.1					
Viola arvensis	37.7								
Silene alba	37.5	16.9	29.3						
Tragopogon dubius	32.8								
Arenaria serpyllifolia	30.7			26					
Anchusa officinalis	30								
Chondrilla juncea	30								
B. Riverside slope of the southern dike (RSD)									
Stellaria media s. str.		72.2							
Equisetum arvense		64.8							
Poa trivialis		62.6							
Arrhenatherum elatius		60.8		25.8					
Galium mollugo		56.8							
Galium aparine		52.9							
Calystegia sepium		52.5							
Dactylis glomerata		49.5							
Lamium purpureum		49.5	41.8						
Ornithogalum umbellatum agg.		48.2				27.2			
Aristolochia clematitis		47.8							
Thalictrum lucidum		47.8							
Lysimachia nummularia		45.4							
Pastinaca sativa subsp. *urens*		42.6							
Galium rubioides		42.5							
Euphorbia virgata		41.2			17.7				
Rumex thyrsiflorus		37.1							
Ranunculus repens		36.8							
Phragmites australis		36.7	20.2						
Centaurea jacea s.l.		35.3							
Clematis integrifolia		35.3			27.9				
Vicia hirsuta		33.8			30.3	19.7			
Taraxacum officinale agg.		33.6			23.8			19	
Pimpinella saxifraga		30							
C. Riverside slope of the northern dike (RND)									
Bromus inermis	20.5		62.2						
Buglossoides arvensis	24.3		58.6	15.7					
Veronica polita	17	38.9	52						
Torilis arvensis			42.7						
Thlaspi perfoliatum		37.4	42.2						
Salvia nemorosa	31.3		38.8	31.3		20			
Falcaria vulgaris			37.7						

Carex melanostachya		34.7		19.8				
Glycyrrhiza echinata		32.4						
Convolvulus arvensis		30.7	23.6	27.2	20.1			
Anthemis ruthenica		30						
Papaver rhoeas		30						
D. Landside slope of the northern dike (LND)								
Centaurea scabiosa s.l.			45.4					
Lepidium draba		26.7	34.6					
Valerianella locusta	27.1	20	30.7	20				
E. Mesotrophic wet meadows								
Myosotis arvensis				67.5				
Cirsium arvense	21.3			63.5				
Ranunculus polyanthemos	33.6			53.1				
Galium verum				49.9	38			
Lathyrus tuberosus	17.5		17.5	46.5				
Carex hirta	26			40.2				
Vicia grandiflora	15.9			39.2				
Amorpha fruticosa				38.6				
Crepis biennis				37.7				
Potentilla reptans				35.3				18.1
Symphytum officinale	20.5			35.3				
Carex praecox	19.3			33.4	15.7			
Linaria vulgaris				30				
F. Closed steppes on loess								
Euphorbia cyparissias					72.1			
Salvia austriaca		18.2			66.3			
Cruciata pedemontana					65.9	25.7		
Valerianella dentata					50.1	37.2		
Eryngium campestre					50			
Crepis pulchra					47.8			
Thesium ramosum					47.8			
Festuca rupicola		21.3	36		47			
Carduus acanthoides					43.8			
Fragaria viridis					42.6			
Hieracium cymosum					42.6			
Achillea collina			32		40.1			
Trifolium campestre				34.1	38.2			
Myosotis ramosissima	18.9		33	22.4	36.6			
Medicago falcata					30			
Potentilla arenaria					30			
G. *Artemisia* salt steppes								
Artemisia santonicum						91.3		
Festuca pseudovina						74.6	38.1	
Podospermum canum						70.7	35.4	
Trifolium striatum						70.7		
Muscari neglectum s.l.						58.4		
Matricaria recutita						57.4		
Bupleurum tenuissimum						56.9		
Allium vineale				39.8		53		
Bromus commutatus						51.8		
Limonium gmelinii subsp. *hungaricum*						51.1	33.9	
Cerastium dubium						47.1	42	
Geranium dissectum				39.7	21.3	47		
Lactuca saligna						42.6		
Sedum caespitosum						42.6		
Plantago tenuiflora						36.8		
Puccinellia limosa						36.8		
Gypsophila muralis						32.8		
Lotus tenuis						30		
H. Salt meadows								
Trifolium micranthum							64.8	

Inula britannica	22.4	56.9	
Trifolium angulatum	49.5	54.5	
Rorippa sylvestris		45.4	
Ranunculus pedatus	31.1	37.9	
Myosotis sicula		37.1	
Taraxacum bessarabicum		37.1	
Polygonum aviculare		36.8	
Oenanthe silaifolia		32.4	
Trifolium retusum	24.3	32.4	
Ranunculus sardous		30.6	
Carex stenophylla		30	
I. Salt marshes			
Eleocharis palustris			76.6
Veronica scutellata			72.2
Agrostis stolonifera			72.1
Lemna minor			64.9
Rorippa austriaca			63.5
Glyceria maxima			56.9
Rumex crispus		16.5	52.3
Lythrum virgatum			47.8
Mentha pulegium		29.7	44.6
Alisma lanceolatum			42.6
Glyceria fluitans			42.6
Schoenoplectus lacustris s.l.			42.6
Beckmannia eruciformis			38.6
Galium palustre agg.			36.8
Persicaria maculosa			36.8
Bolboschoenus maritimus			30
Juncus effusus			30
Ranunculus aquatilis			30
Ranunculus polyphyllus			30